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Soil & Tillage Research 83 (2005) 25–52
Soil & Tillage Research
www.elsevier.com/locate/still

Review

Greenhouse gas contributions and mitigation potential of agricultural practices in northwestern USA and western Canada☆

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Received 12 March 2004; received in revised form 6 January 2005

Abstract

Concern over human impact on the global environment has generated increased interest in quantifying agricultural contributions to greenhouse gas fluxes. As part of a research effort called GRACEnet (Greenhouse Gas Reduction through Agricultural Carbon Enhancement Network), this paper summarizes available information concerning management effects on soil organic carbon (SOC) and carbon dioxide (CO_2), nitrous oxide (N_2O), and methane (CH_4) fluxes in cropland and rangeland in northwestern USA and western Canada, a region characterized by its inherently productive soils and highly variable climate. Continuous cropping under no-tillage in the region increased SOC by 0.27 ± 0.19 Mg C ha^{-1} yr^{-1}, which is similar to the Intergovernmental Panel on Climate Change (IPCC) estimate for net annual change in C stocks from improved cropland management. Soil organic C sequestration potential for rangelands was highly variable due to the diversity of plant communities, soils, and landscapes, underscoring the need for additional long-term C cycling research on rangeland. Despite high variability, grazing increased SOC by 0.16 ± 0.12 Mg C ha^{-1} yr^{-1} and converting cropland or reclaimed mineland to grass increased SOC by 0.94 ± 0.86 Mg C ha^{-1} yr^{-1}. Although there was generally poor geographical coverage throughout the region with respect to estimates of N_2O and CH_4 flux, emission of N_2O was greatest in irrigated cropland, followed by non-irrigated cropland, and rangeland. Rangeland and non-irrigated cropland appeared to be a sink for atmospheric CH_4, but the size of this sink was difficult

Abbreviations: CO_2, carbon dioxide; N_2O, nitrous oxide; CH_4, methane; SOC, soil organic carbon; SIC, soil inorganic carbon; IPCC, Intergovernmental Panel on Climate Change; NT, no-tillage; MT, minimum tillage; CT, conventional tillage; BREB, bowen ratio/energy balance; NDVI, normalized difference vegetation index; CV, coefficient of variation; SE, standard error

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doi:10.1016/j.still.2005.02.008

to determine given the few studies conducted. Researchers in the region are challenged to fill the large voids of knowledge regarding CO_2, N_2O, and CH_4 flux from cropland and rangeland in the northwestern USA and western Canada, as well as integrate such data to determine the net effect of agricultural management on radiative forcing of the atmosphere.

Keywords: GRACEnet; Greenhouse gas flux; Soil organic carbon; North America

Contents

1. Introduction

Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are greenhouse gases that contribute to radiative forcing of the atmosphere. The flux of these gases from agroecosystems is highly dependent on management. Agricultural management can mitigate radiative forcing by increasing soil organic carbon (SOC), decreasing CH_4 and N_2O emissions, or increasing soil CH_4 oxidation (Mosier et al., 2003). Identification of management systems capable of mitigating radiative forcing from agroecosystems will minimize agriculture's impact on the global environment.

A complex network of mechanisms underlie the release and/or uptake of greenhouse gases from agroecosystems. Carbon dioxide is a substrate for photosynthesis, which is the key reaction whereby plants assimilate C, and at least temporarily, sequester it into the landscape (Salisbury and Ross, 1985). Carbon dioxide is also an end product of respiration reactions for both auto- and heterotrophic organisms (Paul and Clark, 1996). Processes of CO_2 uptake and emission are not tightly linked in most terrestrial ecosystems, so fundamental knowledge of both processes is required to understand the dynamics of the C cycle and how it is affected by management and the environment.

Methane can either be released or assimilated into soils, depending on the microbial community and soil moisture conditions (Schutz et al., 1990). Under anaerobic conditions, CH_4 is produced via reduction of CO_2 (Ojima et al., 1993), whereas under aerobic conditions, microbial oxidation of CH_4 causes soils to be sinks for atmospheric CH_4 (Lidstrom and Stirling, 1990). In contrast to CO_2 and CH_4, N_2O flux in agroecosystems is typically unidirectional from soils and plants to the atmosphere. Generation of N_2O in the soil may result from either nitrification or denitrification processes (Firestone and Davidson, 1989).

Overall, rates of soil–atmosphere exchange of N_2O and CH_4 are governed by soil moisture, temperature, substrate supply, and aeration (Mosier et al., 1991; Corre et al., 1996, 1999; Chan and Parkin, 2001).

Numerous reviews of published research have been conducted to estimate greenhouse gas mitigation potential of agricultural management systems. These reviews, mainly at global (Mosier et al., 1996a,b; West and Post, 2002) and national (Lal et al., 1999; Follett et al., 2001a; VandenBygaart et al., 2003) levels, have summarized a wealth of information from long-term experiments in order to refine guidelines developed by the Intergovernmental Panel on Climate Change (IPCC) for greenhouse gas inventories (IPCC, 2000). Although global and national assessments are practical from a sociopolitical standpoint, they tend to integrate information over vastly different ecosystems, thereby ignoring potential regional differences in greenhouse gas flux. Selection of an appropriate spatial resolution to assess greenhouse gas inventories is critical to accurately estimate mitigation potential of agricultural management systems (Adams et al., 2003). Delineation of regions by climatic and edaphic characteristics may be one approach to improve these estimates.

This review of literature focuses on management effects on SOC and greenhouse gas flux for cropland and rangeland in the northwestern USA and western Canada; a region with significant expanses of land used for agriculture. This information will be used to critique IPCC estimates as well as identify knowledge gaps regarding greenhouse gas mitigation for agricultural management systems within the region.

2. Characterization of region

2.1. Geography, climate, soils, native vegetation

The agricultural region reviewed in this paper is delineated by Ecoregion Divisions 330 (Temperate Steppe) and 240 (Marine) in the USA (Bailey, 1995) and the Prairies, Boreal and Tiaga Plains, Boreal, Tiaga, and Montane Cordilleras, and Pacific Maritime Ecozones in Canada (Ecological Stratification Working Group, 1995) (Fig. 1). States and provinces within this region include all or major parts of Kansas, Nebraska, North and South Dakota, Colorado, Wyoming, Montana, Idaho, Oregon, Washington, and Alaska in the USA, and Manitoba, Saskatchewan, Alberta, British Columbia, Yukon, and Northwest Territories in Canada. Although the geographical extent of the region is large (approximately 686 Mha), land area available for agricultural production is restricted to approximately 192 Mha, or 28% of the total (U.S. Census Bureau, 2002; Statistics Canada, 2003). Within the USA, agricultural land in this region occupies about 134 Mha and is found primarily in the Great Plains, Palouse Dry Steppe of eastern Washington and Oregon, and Pacific Lowland of western Oregon and Washington and southeastern Alaska (U.S. Census Bureau, 2002). In Canada, agricultural land in the region occupies about 58 Mha, the majority of which is found in the Prairies and Boreal Plains Ecozones (Statistics Canada, 2003).

The Temperate Steppe Ecoregion of the USA and the Prairies and Boreal Plains Ecozones of Canada have a semiarid to subhumid continental climate, with evaporation typically exceeding precipitation in any given year (Bailey, 1995; Ecological Stratification Working Group, 1995). In general, winters are cold and dry, and summers warm to hot with erratic precipitation. Average annual temperature ranges from −2 °C in the northwest to 18 °C in the southeast. Annual precipitation increases from west to east, with averages ranging from 250 to 760 mm. However, distribution of precipitation throughout the year differs within the region, as the Great Plains receive maximum precipitation during the spring and summer months, while the Palouse region receives most precipitation during the winter months (Rasmussen and Parton, 1994). Although average values for temperature and precipitation provide a glimpse into the climate, the defining climatic characteristic of the region is its variability, as droughts, wet-periods, intense precipitation events, and extreme temperatures are commonplace (Peterson et al., 1996).

The Marine Ecoregion and Pacific Maritime Ecozone are characterized by a humid maritime climate (Bailey, 1995; Ecological Stratification Working Group, 1995). Mild winters and relatively cool summers are typical for the region. Average annual temperature ranges from 5 °C in the north to 13 °C in the south. Precipitation ranges from 1500 to 3000 mm yr^{-1}.

Fig. 1. Approximate boundaries of region. Adapted after Bailey (1995) and Ecological Stratification Working Group (1995).

Soils within the Great Plains and Palouse possess high natural fertility and good moisture-holding capacity. Organic matter accumulation and calcification are the dominant pedogenic processes. Surface soil depths tend to possess increasing organic matter with greater precipitation, while large amounts of precipitated calcium are present at lower depths.

Taxonomically, Great Plains and Palouse soils are Mollisols/Chernozems, although Entisols/Regosols are present throughout the more arid parts of the region. Marine and Pacific Maritime soils are strongly leached and are generally acidic, thereby resulting in the depletion of basic cations from surface horizons. Taxonomically, these soils include Inceptisols/Brunisols, Spodosols/Podzols, Alfisols, and Ultisols (Soil Survey Staff, 1999).

There are six major non-forested ecosystems within the delineated region, each supporting unique native vegetation. Included in the region are tall- and mixed-grass prairies, shortgrass steppe, sagebrush steppe, northern grassland transition zone, and arctic tundra. Remaining remnants of tallgrass prairie in the eastern part of the region are dominated by C_4 grasses (*Andropogon gerardii* and *Sorghastrum nutans*), but contain some C_3 grasses and an assortment of forbs and sedges. Mixed-grass prairies contain a mixture of C_3 and C_4 grasses and forbs, and the component of C_3 and mid-height grasses is greater than found in the shortgrass steppe. Shortgrass steppe is characterized by a mixture of C_3 and C_4 grasses, forbs, and small shrubs, and is often dominated by the C_4 grass *Bouteloua gracilis*. The sagebrush steppe is dominated by a number of shrubs in the *Artemisia* genus, but includes a diversity of other sages and shrubs, with particular plant complexes determined in large part by soil type and moisture availability. The northern

Table 1
Cropland, rangeland, and total surface area for states and provinces within region

State/province	Cropland (Mha)	Rangeland (Mha)	Tame or seeded pasture (Mha)	Other agricultural land (Mha)	Total surface area of state/ province (Mha)
USA[a]					
Colorado	3.6	11.2[b]	NR[c]	NR	27.0
Kansas	10.7	8.5	NR	NR	21.3
Montana	6.1	17.4	NR	NR	38.1
Nebraska	7.9	10.6	NR	NR	20.0
North Dakota	10.1	5.9	NR	NR	18.3
Oregon	1.5	4.8	NR	NR	25.2
South Dakota	6.8	10.4	NR	NR	20.0
Washington	2.7	3.3	NR	NR	17.8
Wyoming	0.9	11.6	NR	NR	25.3
Canada[d]					
Manitoba	5.0	1.6	0.4	0.7	64.8
Saskatchewan	18.5	5.1	1.4	1.2	65.1
Alberta	11.0	6.7	2.2	1.2	66.2
British Columbia	0.7	1.2	0.2	0.5	94.5
Yukon	–	<0.1	–	–	48.2
Northwest Territories	–	<0.1	–	–	134.6
Total	85.5	98.3	4.2	3.6	686.4

[a] Source: U.S. Census Bureau (2002).
[b] Rangeland categorized as 'land in grass' by U.S. Census Bureau (2002).
[c] Not reported.
[d] Source: Statistics Canada (2003).

grassland transition zone represents an ecosystem in Canada and northern Montana between the grassland environments of the south and the boreal forests of the north. In its native state, this zone was characterized by a mosaic of *Populus tremuloides* and fescue grasslands (Padbury et al., 2002). The arctic tundra is characterized by tussocks of *Cladonia rangiferina* and *Eriophorum angustifolium* along with numerous forbs and sedges (Barbour and Billings, 2000). Each of the above ecosystems, if not used for crop production, is grazed by domestic livestock and/or wild animals, and can therefore be considered rangeland (Society for Range Management, 1998). The focus of the discussion on rangeland in this review, however, will be confined to that managed for livestock production.

2.2. Current land use

Land use data were aggregated by state and province, and exclude Alaska and Idaho, which based on the delineated region, contribute little agricultural land to the overall land area. Given these assumptions, cropland and rangeland (including tame or seeded pasture and other agricultural land in Canada) occupy approximately 85 and 106 Mha, respectively (Table 1) (U.S. Census Bureau, 2002; Statistics Canada, 2003). Rangeland area exceeds cropland in all states and provinces except Kansas, North Dakota, Manitoba, Saskatchewan, and Alberta.

Distribution of annual crops within the region varies considerably. Within the northern Great Plains and Palouse, cereal crops such as hard red spring wheat, winter wheat (*Triticum aestivum*L.) and barley (*Hordeum vulgare*L.) are predominant, although oilseed, pulse, and forage crops have increased in planted area significantly since the 1980s (Padbury, 2003). Spring and winter wheat are primary crops within the southern portion of the region, with corn (*Zea mays* L.), sorghum [*Sorghum bicolor* (L.) Moench], proso millet (*Panicum miliaceum* L.), and sunflower (*Helianthus annuus* L.) making up the majority of alternative crops (Westfall et al., 1996). Fallow periods are common throughout the Great

Plains and Palouse, occupying up to 35% of cropland area in any given year (Padbury, 2003). Use of fallow, however, has steadily declined with adoption of conservation tillage (Tanaka et al., 2002). In general, horticultural crops are predominant in the Marine and Pacific Maritime region, although significant area is dedicated to hay and grass seed production (U.S. Census Bureau, 2002).

Rangeland in the region is used exclusively for grazing by cattle, sheep, and wild animals (U.S. Census Bureau, 2002; Statistics Canada, 2003). Approximately 41 million cattle and two million sheep were accounted for in states and provinces in the region in 2001 (Statistics Canada, 2003; NASS, 2004). The number of cattle and sheep utilizing rangeland as a primary source of feed in the region, however, is not known, nor are there estimates regarding the number of cattle that graze crop residue on cropland. Populations of plains bison (*Bison bison* L.) and prairie dogs (*Cynomys ludovicianus*), two animals once predominant throughout the region, are relatively sparse relative to pre-settlement times, but are still present across large private landholdings and national parks (Sims and Risser, 2000).

Table 2
Historical record of soil organic carbon (SOC) loss since conversion to cropping within study region

Location	Textural class	Soil horizon/ depth (cm)	Years under cultivation (yr)	SOC loss		Reference
				Relative (%)	g C kg^{-1} soil	
Brown soil zone	Variable	0–15.2	16	21	6.1	Newton et al. (1945)
Dark brown soil zone	Variable	0–15.2	22	22	8.8	Newton et al. (1945)
Black soil zone	Variable	0–15.2	26	18	11.0	Newton et al. (1945)
Black transition soil zone	Variable	0–15.2	24	24	20.3	Newton et al. (1945)
Gray transition soil zone	Variable	0–15.2	12	12	5.9	Newton et al. (1945)
Gray	Variable	0–15.2	13	27	6.6	Newton et al. (1945)
Lethbridge, AB	L	0–15.2	43	18	3.2	Hill (1954)
Quinton, SK	L	0–10	15	35	19.0	Martel and Paul (1974)
Quinton, SK	L	0–10	60	59	32.0	Martel and Paul (1974)
Hafford, SK	L	0–15	30	45	31.3	Martel and Paul (1974)
Matador, SK	C	0–10	20	19	5.0	Martel and Paul (1974)
Blaine Lake, SK	SiL	A horizon	60	32	15.1	Tiessen et al. (1982)
Blaine Lake, SK	SiL	A horizon	90	58	27.9	Tiessen et al. (1982)
Bradwell, SK	SL	A horizon	70	37	14.0	Tiessen et al. (1982)
Sutherland, SK	C	A horizon	65	46	14.8	Tiessen et al. (1982)
Mandan, ND	SL	0–15.2	30	31	6.6	Hass et al. (1957)
Dickinson, ND	L	0–15.2	40	59	21.3	Hass et al. (1957)
Havre, MT	CL	0–15.2	31	53	9.2	Hass et al. (1957)
Moccasin, MT	CL	0–15.2	39	32	10.5	Hass et al. (1957)
Sheridan, WY	L	0–15.2	30	28	4.7	Hass et al. (1957)
Archer, WY	L	0–15.2	34	41	5.5	Hass et al. (1957)
Akron, CO	SiL	0–15.2	39	46	6.5	Hass et al. (1957)
Colby, KS	SiL	0–15.2	41	45	8.2	Hass et al. (1957)
Hays, KS	SiCL	0–15.2	43	51	12.6	Hass et al. (1957)
Garden City, KS	SL	0–15.2	39	39	4.4	Hass et al. (1957)
Nebraska	Variable	0–30.5	3–7	7	ND[a]	Russel (1929)
Nebraska	Variable	0–30.5	8–15	12	ND	Russel (1929)
Nebraska	Variable	0–30.5	17–30	27	ND	Russel (1929)
Nebraska	Variable	0–30.5	32–44	26	ND	Russel (1929)
Nebraska	Variable	0–30.5	45–60	28	ND	Russel (1929)
North central KS	SiL/SiCL	0–17.8	>30	37	9.1	Hide and Metzger (1939)
South central KS	SiL/SiCL	0–17.8	>30	36	9.6	Hide and Metzger (1939)
Pendleton, OR	SiL	0–20	50	38	9.2	Rasmussen and Albrecht (1998)
Pendleton, OR	SiL	0–20	110	52	12.6	Rasmussen and Albrecht (1998)

[a] ND, not determined.

3. Management effects on soil organic carbon and trace gas flux

3.1. Soil organic carbon in cropland

Historical records indicate significant loss of SOC in the region upon conversion to cropping (Table 2). Average relative loss of SOC was 34 ± 14% (12.1 ± 7.9 g C kg^{-1} soil) for soil depths ≤30 cm. Previous estimates of SOC loss in agricultural lands in the central and northern Great Plains ranged from 20 to 53% (Donigian et al., 1994; Cihacek and Ulmer, 1995; Janzen et al., 1998; VandenBygaart et al., 2003). A plot of data from Table 2 using 15 yr time steps revealed a strong positive relationship between SOC loss and time (Fig. 2). Paustian et al. (1997) observed a similar trend in SOC loss for four wheat-dominated cropping systems, with losses declining rapidly following cultivation and eventually stabilizing with time. The magnitude and rate of SOC loss from cropland is influenced by climate, soil texture, C inputs, degree of soil disturbance, erosion, and SOC status prior to cultivation. Conversely, the maximum increase in soil C storage possible with improved agricultural management will likely be dependent upon similar environmental constraints.

Specific management effects on SOC within the Great Plains and Palouse region are reported in Table 3 (no data were found for the Marine and Pacific Maritime region). Effects of tillage on SOC underscored the value of no-tillage (NT) as a management tool to increase C storage in soil. In western Canada, SOC increased under NT from 0.14 ± 0.13 Mg C ha^{-1} yr^{-1} to 0.53 ± 0.80 Mg C ha^{-1} yr^{-1} with decreasing frequency of fallow (VandenBygaart et al., 2003) (Table 3). Continuous cropping under NT increased SOC by 0.27 Mg C ha^{-1} yr^{-1}. Effects of NT on SOC within a spring wheat–fallow sequence was highly variable across the region, ranging from −0.32 to 0.14 Mg C ha^{-1} yr^{-1}. This finding underscores the importance of interpreting SOC trends within the context of specific soil/climate attributes, as well as taking into account possible impacts of previous management on soil conditions prior to initiating a study.

Soil disturbance by tillage decreased C storage potential within continuous cropping in the region, as the use of minimum tillage (MT) increased SOC by 0.05 Mg C ha^{-1} yr^{-1}, while conventional tillage (CT) decreased SOC (Table 3). Use of tillage (MT or CT) within crop sequences including fallow resulted in consistent SOC losses throughout the region, ranging from −0.08 to −0.53 Mg C ha^{-1} yr^{-1}.

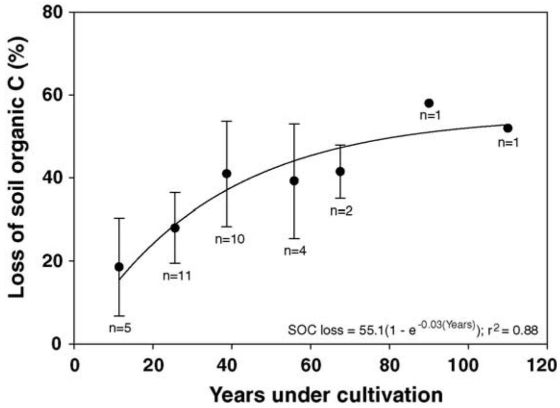

Fig. 2. Plot of soil organic carbon (SOC) loss (%) by years under cultivation for data in Table 2 grouped within 15-yr time steps.

In general, crop sequences resulting in greater input of above- and below-ground biomass to the soil had a positive effect on SOC storage (Table 3). In Colorado, presence of fallow every fourth year increased SOC up to 0.26 Mg C ha^{-1} yr^{-1} relative to sequences with fallow every other year (Ortega et al., 2002). Additional results from Colorado indicated continuous cropping increased SOC by 0.12 Mg C ha^{-1} yr^{-1} compared to crop sequences with fallow, regardless of tillage system used (Bowman et al., 1999). In Oregon, replacing fallow with pea in a wheat rotation resulted in a net increase in SOC storage under MT (0.07 Mg C ha^{-1} yr^{-1}).

In western Canada, addition of forage crops in rotation resulted in significant gains in SOC (Table 3). For instance, replacement of spring wheat–fallow with continuous wheatgrass increased SOC by 0.35 ± 0.19 Mg C ha^{-1} yr^{-1} and inclusion of hay in rotation with fallow and wheat increased SOC by 0.22 ± 0.19 Mg C ha^{-1} yr^{-1} (VandenBygaart et al., 2003). Replacement of wheat with rye (*Secale cereale* L.) increased SOC by 0.10 ± 0.14 Mg C ha^{-1} yr^{-1}, but replacement with flax (*Linum usitatissimum*L.) decreased SOC by 0.15 ± 0.02 Mg C ha^{-1} yr^{-1}. Soil organic C increased by 0.15 ± 0.11 Mg C ha^{-1} yr^{-1} when a green manure crop replaced fallow in a wheat rotation (VandenBygaart et al., 2003).

Table 3
Changes in soil organic carbon under cropland as affected by management within region

Management variable/ location	Textural class	Depth (cm)	Management[a] practice	Duration (yr)	Mean storage rate[b] (Mg C ha^{-1} yr^{-1})	Reference
Tillage						
Western Canada	Variable	Variable	SW–F, NT	14 ± 3.0	0.14 ± 0.13	Multiple studies[c]
Western Canada	Variable	Variable	SW w/F at most every third year, NT	10 ± 0.0	0.23 ± 0.33	Multiple studies[c]
Western Canada	Variable	Variable	Cont. SW, NT	10.7 ± 2.4	0.53 ± 0.80	Multiple studies[c]
Culbertson, MT	SL	0–9	Cont. crop, NT	9	0.22	Pikul and Aase (1995)
Mandan, ND	SiL	0–15.2	SW–WW–SF, NT	12	0.23	Halvorson et al. (2002)
Mandan, ND	SiL	0–15.2	SW–WW–SF, MT	12	0.03	Halvorson et al. (2002)
Mandan, ND	SiL	0–15.2	SW–WW–SF, CT	12	(0.14)	Halvorson et al. (2002)
Pendleton, OR	SiL	0–20	Cont. WW, NT	10	0.08	Rasmussen and Albrecht (1998)
Pendleton, OR	SiL	0–20	Cont. WW, CT	60	(0.01)	Rasmussen and Albrecht (1998)
Pendleton, OR	SiL	0–20	WW–F, MT	40	(0.08)	Rasmussen and Albrecht (1998)
Pendleton, OR	SiL	0–20	WW–F, CT	40	(0.08)	Rasmussen and Albrecht (1998)
Sidney, NE	SiL/L	0–30	SW–F, NT	22–27	(0.32)	Doran et al. (1998)
Sidney, NE	SiL/L	0–30	SW–F, CT	22–27	(0.53)	Doran et al. (1998)
Crop sequence						
Western Canada	Variable	Variable	Cont. SW	23.0 ± 8.8	0.15 ± 0.06	Multiple studies[c]
Western Canada	Variable	Variable	SW–F, replaced w/cont. wheatgrass	7.3 ± 2.6	0.35 ± 0.19	Multiple studies[c]
Western Canada	Variable	Variable	SW replaced with flax	16 ± 1.6	(0.15 ± 0.02)	Multiple studies[c]
Western Canada	Variable	Variable	Hay w/ SW–F	31.3 ± 4.5	0.22 ± 0.19	Multiple studies[c]
Western Canada	Variable	Variable	F replaced with legume green manure in F–SW–SW	20.6 ± 8.2	0.15 ± 0.11	Multiple studies[c]
Western Canada	Variable	Variable	SW replaced with rye in SW–F	19.3 ± 6.6	0.10 ± 0.14	Multiple studies[c]
Akron, CO	L	0–15	Plots w/ F	4	0.18	Bowman et al. (1999)
Akron, CO	L	0–15	Plots w/o F	4	0.30	Bowman et al. (1999)
Sterling, CO	L	0–10	SW–F	8	(0.20)	Ortega et al. (2002)
Sterling, CO	L	0–10	SW–C–M–F	8	(0.10)	Ortega et al. (2002)
Stratton, CO	SiCL	0–10	SW–F	8	0.17	Ortega et al. (2002)
Stratton, CO	SiCL	0–10	SW–C–M–F	8	0.43	Ortega et al. (2002)
Walsh, CO	SCL	0–10	SW–F	8	0.05	Ortega et al. (2002)
Walsh, CO	SCL	0–10	SW–C–M–F	8	0.05	Ortega et al. (2002)
Pendleton, OR	SiL	0–20	WW–F replaced with WW–P, CT	28	(0.03)	Rasmussen and Albrecht (1998)

Table 3 (*Continued*)

Management variable/ location	Textural class	Depth (cm)	Management[a] practice	Duration (yr)	Mean storage rate[b] (Mg C ha^{-1} yr^{-1})	Reference
Pendleton, OR	SiL	0–20	WW–F replaced with WW–P, MT	28	0.07	Rasmussen and Albrecht (1998)
Fertilization						
Canada	Variable	Variable	Variable N rates	23.2 ± 4.5	0.23 ± 0.13	Multiple studies[c]
Akron, CO	SiL	0–15	B–C–Oat/Pea, 45 kg N ha^{-1}	11	0.09	Halvorson et al. (1999)
Akron, CO	SiL	0–15	B–C–Oat/Pea, 67 kg N ha^{-1}	11	0.14	Halvorson et al. (1999)
Akron, CO	SiL	0–15	B–C–Oat/Pea, 134 kg N ha^{-1}	11	0.18	Halvorson et al. (1999)
Pendleton, OR	SiL	0–30	WW–F, 0 kg N ha^{-1}	55	(0.19)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, 45 kg N ha^{-1}	55	(0.16)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, 90 kg N ha^{-1}	55	(0.13)	Rasmussen and Parton (1994)
Manure and residue management						
Pendleton, OR	SiL	0–20	WW–F, manure applied at 11.2 MT ha^{-1} yr^{-1}	56	0.02	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, Pea vine applied at 1.12 MT ha^{-1} yr^{-1}	55	(0.10)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, nB-N_0	55	(0.19)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, fB-N_0	55	(0.21)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, sB-N_0	55	(0.17)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, nB-N_{45}	34	(0.05)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, nB-N_{90}	34	(0.04)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, sB-N_{45}	22	(0.03)	Rasmussen and Parton (1994)
Pendleton, OR	SiL	0–30	WW–F, sB-N_{90}	22	0.01	Rasmussen and Parton (1994)
Irrigation						
Lethbridge, AB	CL	0–30	SW–SGB–SW–AL, MT	4	1.68	Hao et al. (2001a)
Lethbridge, AB	CL	0–30	SW–SGB–SW–AL, CT	4	0.70	Hao et al. (2001a)
Sandhills region, NE	S/LS	0–30	Cont. C	15	0.11	Lueking and Schepers (1985)
Ft. Collins, CO	CL	0–15.2	Cont. C, NT	4	1.44	Halvorson et al. (2003)
Ft. Collins, CO	CL	0–15.2	Cont. C, CT	4	0.03	Halvorson et al. (2003)

[a] NT, no-tillage; MT, minimum tillage; CT, conventional tillage; B, barley; SW, spring wheat; F, fallow; WW, winter wheat; SF, sunflower; C, corn; M, millet; Cont., continuous; SGB, sugar beet; AL, annual legume; P, pea; N_0, 0 kg N ha^{-1}; N_{45}, 45 kg N ha^{-1}; N_{90}, 90 kg N ha^{-1}; fB, fall burn; sB, spring burn; nB, not burned.

[b] Values in parentheses indicate SOC loss.

[c] Multiple studies summarized by VandenBygaart et al. (2003) including Campbell et al. (1996a), Campbell et al. (1996b), Campbell et al. (1995), Larney et al. (1997), Nyborg et al. (1995), Campbell et al. (2001), Grant and LaFond, 1994, McConkey et al. (2003), Hao et al. (2001a), Elliott and Efetha (1999); Dormaar and Carefoot (1998), Franzluebbers and Arshad (1996), Miller et al. (1999), Angers et al. (1997), Campbell et al. (1991), Bremer et al. (1994), Campbell et al. (1998), Biederbeck et al. (1998), Campbell et al. (2000a), Campbell et al. (2000b), Campbell and Zentner (1993), Bremer et al. (2002), Nyborg et al. (1998), Nyborg et al. (1999), Malhi et al. (1997), Solberg et al. (1997), Liang and Mackenzie (1992), Gregorich et al. (1996), Malhi et al. (2002), and Izaurralde et al. (2001).

Given the positive relationships among soil fertility status, biomass production, and SOC content (Parton et al., 1996), N fertilizer should increase SOC. The summary by VandenBygaart et al. (2003) supported this hypothesis, as application of inorganic N at varying rates increased SOC by 0.23 ± 0.13 Mg C ha^{-1} yr^{-1} for studies conducted in Canada (Table 3). Nitrogen fertilizer effects on SOC throughout the rest of the region varied with cropping frequency. Halvorson et al. (1999) found that SOC increased from 0.09 to 0.18 Mg C ha^{-1} yr^{-1} with increasing N rate for continuous cropping under NT management in Colorado (Table 3). Conversely, loss of SOC occurred for a wheat–fallow crop sequence across a range of N rates in Oregon (Table 3). Overall, a good N fertility program appeared to increase SOC in continuously cropped management systems, although a full accounting of manufactured N on CO_2 emission would be necessary to determine the net effect of N-fertilization on C balance in these cropping systems.

Application of manure as organic fertilizer has been promoted as a management approach to increase SOC (Lal et al., 1999). VandenBygaart et al. (2003) calculated that manure of varying types and rates increased SOC by 28% (relative) over a 13-yr period in Canada (data not shown). Most studies in their summary reported SOC on a concentration basis only. Within the USA, application of cattle manure and straw at 11.2 MT ha^{-1} yr^{-1} resulted in a slight increase (0.02 Mg C ha^{-1} yr^{-1}) in SOC over a 56-yr period in Oregon (Table 3). While increased SOC from manure application is possible, this management practice essentially transfers C from one system to another.

Few studies have investigated the effects of crop residue removal on SOC in the northwestern USA and western Canada. Whether wheat straw was burned or not had variable effects on SOC depending upon fertilization regime (Table 3). Similarly, few studies have evaluated SOC dynamics under irrigated agriculture in the region. Soil organic C increased 0.79 ± 0.75 Mg C ha^{-1} yr^{-1} with irrigation in five studies (Table 3). Reducing soil disturbance (i.e., MT or NT) combined with irrigation resulted in SOC increases on the high end of this variation.

No known research has been conducted within the region on soil inorganic carbon (SIC) changes under cropland. Loss of SIC would be possible through soil acidification, while SIC gain could occur from Ca^{2+} illuviation to the subsoil, resulting in the formation of $CaCO_3$ (Monger and Martinez-Rios, 2001). This latter mechanism has been proposed to occur under long-term cropping systems in North Dakota (Cihacek and Ulmer, 1995; Liebig et al., 2002), but has yet to be quantified.

3.2. Trace gas flux in cropland

Several studies in the region have quantified N_2O, CH_4, and CO_2 flux from cropland. Typically, trace gas flux has been measured using closed chamber techniques within cropping systems (Mosier, 1989). However, less common and more technically demanding techniques, such as micrometeorological approaches using flux towers or aircraft have been used (Matson and Harriss, 1988; Edwards et al., 2003). In this review, all studies used closed chamber techniques to determine N_2O and CH_4 flux, while two studies used a micrometeorological method to determine CO_2 flux.

Flux of N_2O, CH_4, and CO_2 have been determined in the region with variations in N-fertilization, tillage and residue management, manure type and application rate, and nitrification inhibitors. Due to specialized research expertise, studies have been clustered in northeastern Colorado, southwestern Saskatchewan, central and southern Alberta, and Alaska.

3.2.1. N_2O emission

Nitrous oxide emission from dryland cropping systems within the region varied from 1.0 to 7.1 g N ha^{-1} d^{-1} (Table 4). Emission of N_2O from cropland has been similar or slightly greater than from native grassland or grass sod, depending on the location and specific management practices employed (Kessavalou et al., 1998a; Mosier et al., 2003). Greater intensity of tillage and inclusion of fallow have tended to increase N_2O emission within the region (Table 4). Kessavalou et al. (1998a) found lower N_2O emission under NT than moldboard plow, although the difference was small (0.3 g N ha^{-1} d^{-1}). Both Mosier et al. (1991) and Lemke et al. (1999) observed greater N_2O emission from cropland under fallow than unfertilized wheat, presumably due to wetter soil conditions. Nitrogen fertilization under dryland

Table 4
Management effects on N_2O emission in cropland within study region

Management variable/ Location	Management[a] practice	Measurement[b] period	N_2O emission[c] (g N ha^{-1} d^{-1})	Reference
Dryland (non-irrigated)				
Nunn, CO	WW, No fertilizers or pesticides	Annual (90)	2.6	Mosier et al. (1991)
	F, No fertilizers or pesticides	Annual (90)	4.5	Mosier et al. (1991)
	Native grassland	Annual (90)	3.5	Mosier et al. (1991)
Sterling, CO	SW–C–F, NT, 112 kg N ha^{-1}, mid-slope	Annual (02–03)	2.1	Mosier et al. (2003)
	SW–C–F, NT, 112 kg N ha^{-1}, toeslope	Annual (02–03)	1.8	Mosier et al. (2003)
	Cont. crop, NT, 112 kg N ha^{-1}, mid-slope	Annual (02–03)	1.9	Mosier et al. (2003)
	Cont. crop, NT, 112 kg N ha^{-1}, toeslope	Annual (02–03)	2.3	Mosier et al. (2003)
	Native grassland, mid-slope	Annual (02–03)	1.0	Mosier et al. (2003)
	Native grassland, toeslope	Annual (02–03)	0.9	Mosier et al. (2003)
Sidney, NE	WW–F, NT, No fertilizers	Annual (93–95)	1.1	Kessavalou et al. (1998a)
	WW–F, subtill, No fertilizers	Annual (93–95)	1.3	Kessavalou et al. (1998a)
	WW–F, plow, No fertilizers	Annual (93–95)	1.4	Kessavalou et al. (1998a)
	Grass sod	Annual (93–95)	0.9	Kessavalou et al. (1998a)
Breton, AB	Wheat, 56 kg N ha^{-1}	Annual (95)	2.5 (1.3%)	Lemke et al. (1998)
Cooking Lake, AB	Wheat, 100 kg N ha^{-1}	Annual (95)	5.5 (1.7%)	Lemke et al. (1998)
Ellerslie, AB	Wheat, 56 kg N ha^{-1}	Annual (95)	6.8 (1.7%)	Lemke et al. (1998)
Josephburg, AB	Wheat, 100 kg N ha^{-1}	Annual (95)	5.5 (1.5%)	Lemke et al. (1998)
Eckville, AB	Barley, 25 kg N ha^{-1}	Annual (95)	7.1 (1.6%)	Lemke et al. (1998)
Ellerslie, AB	Wheat, 56 kg N ha^{-1}, Intensive tillage	Spring, growing season (94–95)	2.1[d] (0.2–2.0%)	Lemke et al. (1999)
Ellerslie, AB	Wheat, 0 kg N ha^{-1}, Intensive tillage	Spring, growing season (94–95)	1.5[d]	Lemke et al. (1999)
Ellerslie, AB	Fallow, intensive tillage	Spring, growing season (94–95)	1.8[d]	Lemke et al. (1999)
Ellerslie, AB	Wheat, 56 kg N ha^{-1}, NT	Spring, growing season (94–95)	1.2[d] (0.7–2.0%)	Lemke et al. (1999)
Ellerslie, AB	Wheat, 0 kg N ha^{-1}, NT	Spring, growing season (94–95)	0.5[d]	Lemke et al. (1999)
Ellerslie, AB	Fallow, NT	Spring, growing season (94–95)	1.1[d]	Lemke et al. (1999)
Delta Junction, AK	Barley, 90 kg N ha^{-1}, Straw retained	Snow-free period (94)	5.0	Cochran et al. (1997)
Delta Junction, AK	Barley, 90 kg N ha^{-1}, Straw removed	Snow-free period (94)	4.0	Cochran et al. (1997)
Delta Junction, AK	Barley, 12 kg N ha^{-1}, Straw retained	Snow-free period (94)	1.5	Cochran et al. (1997)
Delta Junction, AK	Barley, 12 kg N ha^{-1}, Straw removed	Snow-free period (94)	1.0	Cochran et al. (1997)
Irrigation				
Berthold, CO	Corn, 200 kg N ha^{-1}	Growing season (78)	19.8 (1.3%)	Mosier and Hutchinson (1981)
Ft. Collins, CO	Corn, 200 kg N ha^{-1}	Growing season (82)	24.6 (1.5%)	Mosier et al. (1986)
	Corn, 0 kg N ha^{-1}	Growing season (82)	18.6	Mosier et al. (1986)

Table 4 (*Continued*)

Management variable/ Location	Management[a] practice	Measurement[b] period	N_2O emission[c] (g N ha^{-1} d^{-1})	Reference
	Barley, 200 kg N ha^{-1}	Growing season (83)	7.6 (0.4%)	Mosier et al. (1986)
	Barley, 0 kg N ha^{-1}	Growing season (83)	4.5	Mosier et al. (1986)
Lethbridge, AB	Canola, 0 kg N ha^{-1}, No straw, Fall plow	Annual (96–97)	1.2	Hao et al. (2001b)
	Canola, 0 kg N ha^{-1}, No straw, Direct seed	Annual (96–97)	1.6	Hao et al. (2001b)
	Canola, 0 kg N ha^{-1}, Straw retained, Fall plow	Annual (96–97)	5.2	Hao et al. (2001b)
	Canola, 100 kg N ha^{-1}, no straw, fall plow	Annual (96–97)	15.6	Hao et al. (2001b)
	Canola, 100 kg N ha^{-1}, no straw, direct seed	Annual (96–97)	5.7	Hao et al. (2001b)
	Canola, 100 kg N ha^{-1}, no straw, spring plow	Annual (96–97)	4.6	Hao et al. (2001b)
	Canola, 100 kg N ha^{-1}, straw retained, fall plow	Annual (96–97)	8.6	Hao et al. (2001b)
	Canola, 100 kg N ha^{-1}, straw retained, spring plow	Annual (96–97)	2.5	Hao et al. (2001b)
Manure management				
Lethbridge, AB	Cattle manure, 0 Mg ha^{-1}, irrigated barley	Annual (93–94)	1.9	Chang et al. (1998)
	Cattle manure, 60 Mg ha^{-1}, irrigated barley	Annual (93–94)	30.1 (2–4%)	Chang et al. (1998)
	Cattle manure, 120 Mg ha^{-1}, irrigated barley	Annual (93–94)	63.0 (2–4%)	Chang et al. (1998)
	Cattle manure, 180 Mg ha^{-1}, irrigated barley	Annual (93–94)	153.3 (2–4%)	Chang et al. (1998)
Greeley, CO	Sewage sludge, 0 Mt ha^{-1}, irrigated barley	Growing season (79)	5.3 (0.4%)	Mosier et al. (1982)
	Sewage sludge, 16.7 Mt ha^{-1}, irrigated barley	Growing season (79)	7.0 (0.8%)	Mosier et al. (1982)
	Sewage sludge, 83.5 Mt ha^{-1}, irrigated barley	Growing season (79)	27.0 (1.0%)	Mosier et al. (1982)
Fertilizer management				
Greeley, CO	0 kg N ha^{-1}, irrigated barley	Growing season (79)	3.4	Mosier et al. (1982)
	56 kg N ha^{-1}, NH_4NO_3, irrigated barley	Growing season (79)	5.9 (0.7%)	Mosier et al. (1982)
	112 kg N ha^{-1}, NH_4NO_3, irrigated barley	Growing season (79)	6.7 (0.4%)	Mosier et al. (1982)
	224 kg N ha^{-1}, NH_4NO_3, irrigated barley	Growing season (79)	9.2 (0.4%)	Mosier et al. (1982)
Nitrification inhibitors				
Ft. Collins, CO	Urea, 218 kg N ha^{-1}, irrigated corn	Growing season (89–90)	25.8 (0.7–1.5%)	Bronson et al. (1992)
	Urea + nitripyrin, 218 kg N ha^{-1}, irrigated corn	Growing season (89–90)	11.1	Bronson et al. (1992)
	Urea + Encap. CaC_2, 218 kg N ha^{-1}, irrigated corn	Growing season (89–90)	14.0	Bronson et al. (1992)
Ft. Collins, CO	Urea, 90 kg N ha^{-1}, irrigated barley	Growing season (93)	8.2 (<0.5%)	Delgado and Mosier (1996)
	Urea + dicyandiamide, 90 kg N ha^{-1}, irrigated barley	Growing season (93)	8.2	Delgado and Mosier (1996)

Table 4 (*Continued*)

Management variable/ Location	Management[a] practice	Measurement[b] period	N_2O emission[c] (g N ha^{-1} d^{-1})	Reference
	Polyolefin coated urea, 90 kg N ha^{-1}, irrigated barley	Growing season (93)	6.9	Delgado and Mosier (1996)
	Check, 0 kg N ha^{-1}, irrigated barley	Growing season (93)	4.5	Delgado and Mosier (1996)

[a] NT, no-tillage; SW, spring wheat; F, fallow; WW, winter wheat; C, corn; Cont., continuous.
[b] Values in parentheses indicate year(s) study was conducted.
[c] Values in parentheses represent the percentage of N_2O emitted per unit of N input.
[d] Values represent cumulative N_2O emissions during measurement period and are expressed as kg N ha^{-1}.

conditions increased N_2O emission from 40 to 300% in Alberta and the subacrtic region of Alaska (Table 4) (Cochran et al., 1997; Lemke et al., 1999).

Unlike dryland cropping conditions where N_2O emission rarely exceeds 5 g N ha^{-1} d^{-1}, N_2O emission from irrigated cropland can be much greater. Mosier and Hutchinson (1981) and Mosier et al. (1986) observed N_2O emission of 18.6–24.6 g N ha^{-1} d^{-1} from irrigated corn in Colorado (Table 4). Emission of N_2O from irrigated barley was $\leq$30% of that from corn (Mosier et al., 1986), presumably due to differences in growth and developmental patterns between crops, as well as lower soil temperatures during the measurement period for barley. Hao et al. (2001b) observed N_2O emission of 1.2–5.2 g N ha^{-1} d^{-1} without fertilization and rates of 2.5–15.6 g N ha^{-1} d^{-1} with 100 kg N ha^{-1} for irrigated canola (Table 4). Removal of crop residue resulted in increased N_2O emission by 0.3 g N ha^{-1} d^{-1} when averaged across fertilization and tillage treatments. Additionally, fall plowing increased N_2O emission by 4.1 g N ha^{-1} d^{-1} as compared with spring plowing within the same residue treatments due to greater N mineralization and warmer soil temperatures.

Manure and other organic amendments in crop production provide organic matter and plant nutrients to improve crop production (Chang et al., 1998). However, when applied excessively, manure can have deleterious effects on air and water quality (Chang and Entz, 1996; Lessard et al., 1996). Increasing application rates of either cattle manure (Chang et al., 1998) or sewage sludge (Mosier et al., 1982) resulted in consistent and substantial increases in N_2O emission from 30.1 to 153.3 g N ha^{-1} d^{-1} (Table 4).

The effect of inorganic N-fertilization on N_2O emission has been well documented (Eichner, 1990). Averaged across five studies with multiple fertilizer rates, N_2O emission was 4.2 g N ha^{-1} d^{-1} without fertilizer and increased 3.2 g N ha^{-1} d^{-1} with every 100 kg N ha^{-1} applied (Table 4).

Nitrification inhibitors, which act to slow the oxidation of NH_4-N to NO_3-N, could limit the amount of NO_3-N available for denitrification. Bronson et al. (1992) observed that nitripyrin and encapsulated calcium carbide reduced N_2O emission by 57 and 46%, respectively, when applied with urea on irrigated corn (Table 4). Polyolefin-coated urea also reduced N_2O emission under irrigated conditions (Delgado and Mosier, 1996).

Nitrous oxide emission from cropland can vary significantly with time. In the Parkland region of Alberta, Lemke et al. (1998) found 16–60% of estimated annual N_2O emission to occur during spring thaw. Similarly, Nyborg et al. (1997) found that nearly all annual N_2O emission occurred within two weeks during and after spring thaw. In Colorado, N_2O emission was 3.8 g N ha^{-1} d^{-1} prior to irrigation of corn and 550 g N ha^{-1} d^{-1} two days after irrigation (Mosier and Hutchinson, 1981). A five-fold increase in N_2O emission occurred after wetting the soil surface in a winter wheat field in western Nebraska (Kessavalou et al., 1998b). These findings underscore the importance of intensive sampling schemes during spring thaw and immediately after irrigation and/or precipitation events.

3.2.2. CH_4 flux

Cropping systems in semiarid regions have been proposed as potential net sinks for atmospheric CH_4 (Paustian et al., 1995), although limited research has been conducted to quantify this potential. Across all dryland cropping system studies, CH_4 uptake was 3.8 ± 2.3 g C ha^{-1} d^{-1}, suggesting that agricultural soils in the region could indeed be a sink for CH_4

Table 5
Management effects on CH_4 uptake in cropland within study region

Location	Management[a] practice	Measurement[b] period	CH_4 uptake ($g\ C\ ha^{-1}\ d^{-1}$)	Reference
Nunn, CO	WW, no fertilizers or pesticides	Annual (90)	1.3	Mosier et al. (1991)
	F, no fertilizers or pesticides	Annual (90)	1.8	Mosier et al. (1991)
	Native grassland	Annual (90)	2.6	Mosier et al. (1991)
Sidney, NE	WW–F, NT, no fertilizers	Annual (93–95)	7.7	Kessavalou et al. (1998a)
	WW–F, subtill, no fertilizers	Annual (93–95)	7.2	Kessavalou et al. (1998a)
	WW–F, plow, no fertilizers	Annual (93–95)	6.8	Kessavalou et al. (1998a)
	Grass sod	Annual (93–95)	8.7	Kessavalou et al. (1998a)
Delta Junction, AK	Barley, disked fall and spring	Snow-free period (94)	2.3	Cochran et al. (1997)
	Barley, disked spring	Snow-free period (94)	2.9	Cochran et al. (1997)
	Barley, NT	Snow-free period (94)	3.2	Cochran et al. (1997)
	Barley, straw removed	Snow-free period (94)	2.7	Cochran et al. (1997)
	Barley, straw retained	Snow-free period (94)	3.0	Cochran et al. (1997)
	Barley, 12 kg N ha^{-1}	Snow-free period (94)	3.1	Cochran et al. (1997)
	Barley, 90 kg N ha^{-1}	Snow-free period (94)	2.6	Cochran et al. (1997)
Fairbanks, AK	Grassland, level topography, 0 kg N ha^{-1}	Snow-free period (92)	2.6	Cochran et al. (1995) and Mosier et al. (1997)
	Grassland, level topography, 90 kg N ha^{-1}	Snow-free period (92)	2.8	Cochran et al. (1995) and Mosier et al. (1997)
	Grassland, 15% slope, 0 kg N ha^{-1}	Snow-free period (92)	5.2	Cochran et al. (1995) and Mosier et al. (1997)
	Grassland, 15% slope, 90 kg N ha^{-1}	Snow-free period (92)	4.5	Cochran et al. (1995) and Mosier et al. (1997)

[a] WW, winter wheat; F, fallow; NT, no-tillage.
[b] Values in parentheses indicate year(s) study was conducted.

(Table 5). Conversion of native vegetation to cropping appears to reduce CH_4 uptake by 50% (Mosier et al., 1991). Soil uptake of CH_4 was 8.7 g C ha^{-1} d^{-1} under grass sod, 7.7 g C ha^{-1} d^{-1} under NT, and 6.8 g C ha^{-1} d^{-1} under moldboard plow in western Nebraska (Kessavalou et al., 1998a). Cochran et al. (1997) also found that increased tillage intensity decreased CH_4 uptake within the subarctic region of Alaska. Fertilization with N generally reduced CH_4 uptake by 0.3 ± 0.5 g C ha^{-1} d^{-1}, possibly from the interference of excess NH_4^+ on the activity of methanotrophic bacteria in soil (Bronson and Mosier, 1994).

3.2.3. CO_2 flux

Measurement of CO_2 flux within cropland provides insight on short-term C dynamics that otherwise cannot be assessed through quantification of SOC. Kessavalou et al. (1998a) observed lower CO_2 emission from NT (7.6 kg C ha^{-1} d^{-1}) than from moldboard plow (10.4 kg C ha^{-1} d^{-1}) for a wheat–fallow system in western Nebraska. Similarly, Curtin et al. (2000) found 20–25% lower CO_2 emission under NT than under CT in continuous wheat; a difference attributed to greater physical release of soil CO_2 under CT and slower decomposition of crop residues under NT. Although tillage resulted in greater soil CO_2 emission, Ellert and Janzen (1999) estimated 3.6–7.2 kg C ha^{-1} were lost from the soil as CO_2 during individual tillage events in southern Alberta, which when compared to the annual emission of CO_2 within a wheat–fallow system represented <5% of total C loss.

Annual C dynamics varies with cropping intensity. Inclusion of fallow, in general, results in lower CO_2 emission relative to treatments cropped to barley or wheat (Akinremi et al., 1999; Curtin et al., 2000). Despite lower CO_2 emission during fallow than during cropping, overall C balance of crop–fallow systems is often negative depending on the type of tillage employed. Using a combination of micrometeorological and chamber methods, McGinn and Akinremi (2001) estimated annual C loss from a barley–fallow system of 0.41–0.60 Mg C ha^{-1}. In continuous wheat under NT, Curtin et al. (2000) observed an increase in SOC of 0.40 Mg C ha^{-1} yr^{-1}.

3.3. Soil organic carbon in rangeland

Native rangelands could be a significant C source or sink because of their high C density and vast land area they occupy (Schuman et al., 2002). Although rates of change in system C are generally slow compared to other agricultural systems, the geographical breadth of rangelands makes them a significant component of global C flows, and therefore critically important to greenhouse gas mitigation efforts (Follett, 2001).

Two characteristics of rangeland ecosystems are: (1) high degree of spatial and temporal variability in soil properties (Bird et al., 2002), and (2) ecosystem C storage predominately (>90%) in soil organic matter (Burke et al., 1997). Rangelands contain a variety of plant species and growth forms (grasses, herbs, trees, and shrubs) that respond in different ways and at different rates to environmental and management inputs (Polley et al., 2000). Soil organic C is spatially variable, not only horizontally across the landscape (Burke et al., 1999), but also vertically within the soil profile, where SOC tends to be correlated with root mass and distribution (Reeder et al., 2001a). This inherent variability not only increases difficulty in quantifying the mass and distribution of soil C (Bird et al., 2002), but also increases difficulty of discerning the effects of management from those of the environment and environmental change (Polley et al., 2000).

Compared with croplands, relatively few studies have been conducted on SOC in native rangelands. Unfortunately, of the studies conducted, many have reported C concentration rather than mass (data not shown), which decreases the utility of the data for estimating SOC reserves and response to changes in the environment or management (Reeder et al., 2001a).

Levels of SOC in rangeland generally tend to increase with precipitation as the result of increasing

Table 6
Evaluations of soil organic carbon (SOC) in rangeland ecosystems[a]

Ecosystem/location	Soil classification	Soil horizon/ depth (cm)	SOC (mass-conc)	Mean storage rate (Mg C ha^{-1} yr^{-1})	Management/ treatment	Reference
Studies within rangeland ecosystems						
Mixed-grass prairie						
North Dakota	Typic Haploboroll	107	Mass-conc	0.29	Stocking rate	Frank et al. (1995) and Frank (2004)
South Dakota	Variable	100	Mass	0.33–1.56	Interseeded legume	Mortenson et al. (2004)
Wyoming	Variable	15	Mass	1.94	Reclaimed mine land	Stahl et al. (2003)
Shortgrass steppe (Central Plains Experimental Range, Nunn, CO, and Pawnee National Grasslands)						
	Ustollic Haplargid	90	Mass	0.07	Stocking rate	Reeder et al. (2004) and Reeder and Schuman (2002)
Tallgrass prairie (Konza prairie, Kansas)						
	Unknown	15	Mass	0.22	Fire and grazing	Rice (2000)
	Unknown	15	Mass	1.6	N-fertilization	Rice (2000)
Studies across rangeland ecosystems						
MGP, SS[b]	Variable	30	Mass	0.12	Grazed vs. ungrazed, climate gradient	Derner et al. (1997)
SGS, SS	Variable	30	Mass	0.4–1.16	Native vs. crop vs. CRP	Reeder et al. (1998)
TGP, SGS, MGP	Variable	100	Mass	0.91	Native vs. crop vs. CRP	Follett et al. (2001b)

[a] A more complete table including 57 references may be obtained by contacting the author.
[b] MGP, mixed-grass prairie; SGS, shortgrass step; TGP, tallgrass prairie; SS, sagebrush steppe.

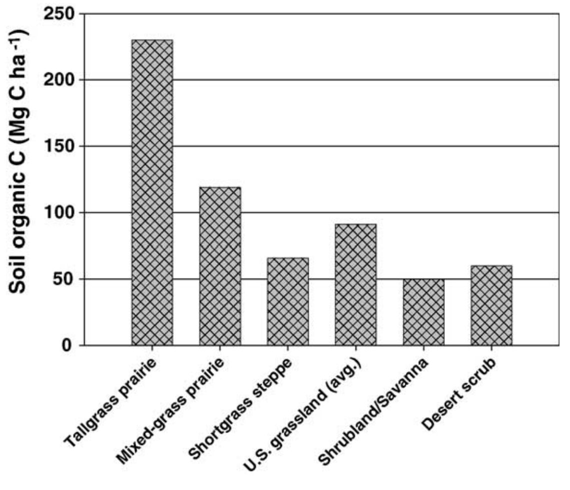

Fig. 3. Soil organic carbon (SOC) to 1 m in different rangeland ecosystems in North America. Values for each ecosystem adapted from the following publications: tallgrass prairie, Lal (2001); mixed-grass prairie, Schuman et al. (1999) and unpublished data; shortgrass steppe, Reeder et al. (2004); average grassland, Follett (2001); shrubland/Savanna and desert scrub, Paul and Clark (1996).

plant productivity, and decrease with temperature as the result of increasing decomposition rates (Burke et al., 1989a). Thus, Aridisols usually have the lowest levels of SOC, while Mollisols (prairie grassland soils) have the highest (Brady and Weil, 1999). In rangeland soil profiles to 1 m, SOC averages 50 Mg C ha^{-1} in the shrub-grass plant communities to 230 Mg C ha^{-1} in the tallgrass prairie (Fig. 3). Average SOC across the historic native grassland region of the USA has been estimated at 91.5 Mg C ha^{-1} to 1 m (Follett et al., 2001a), and 123 ± 48 Mg C ha^{-1} to 2 m (Follett, 2001). Data in Follett (2001) demonstrate the high variability in SOC across different rangeland ecosystems, but SOC variability is also high within an ecosystem. For example, SOC (1 m) in the Konza tallgrass prairie of Kansas ranges from 150 to 300 Mg C ha^{-1}, depending on landscape position (Lal, 2001).

Within the past 10–15 yr, a number of studies have assessed the role of grazing and other management practices such as fire and fertilizer N application on the SOC balance of various rangeland ecosystems (Schuman et al., 2002). Estimates of grazing-induced changes in SOC have been variable (Milchunas and Lauenroth, 1993; Schuman et al., 2001). For example, grazing of rangelands has been shown to increase SOC mass (Dormaar et al., 1994; Derner et al., 1997; Schuman et al., 1999; Reeder, 2003), decrease SOC mass (Bauer et al., 1987; Frank et al., 1995), and have no significant effect on SOC mass (Dormaar et al., 1977; Kieft, 1994; Willms et al., 2002; Haferkamp and MacNeil, 2004). Inconsistencies in SOC response to livestock grazing are due to a number of factors including differences in sampling procedures, climate, plant community composition, and grazing history (Milchunas and Lauenroth, 1993; Schuman et al., 1999; Reeder and Schuman, 2002), as well as differences in seasonal grazing frequency, intensity, and duration (Naeth et al., 1991; Frank et al., 1995; Reeder et al., 2001b). Lower levels of SOC with grazing may have been due to reduced litter inputs relative to ungrazed exclosures (Burke et al., 1997). Higher levels of SOC with grazing have, in many instances, been associated with shifts in the plant community to increased abundance of perennial grasses with high root-to-shoot ratios (Dormaar and Willms, 1990; Frank et al., 1995; Schuman et al., 1999; Reeder et al., 2004). Other factors contributing to increased SOC storage with well-managed grazing include stimulation of C cycling from aboveground plant litter to soil (Schuman et al., 1999), stimulation of aboveground production (Frank and McNaughton, 1993), increased tillering and rhizome production (Floate, 1981; Schuman et al., 1990), and possible stimulation of root exudation and respiration rates (Dyer and Bokhari, 1976).

Management alternatives for arid and semi-arid native rangelands tend to be limited to modifications of stocking rate, duration and season of grazing, and introduction of legumes into the system (Mortenson et al., 2004). Increases in SOC storage with improved management would therefore likely be slow, estimated at 0.10–0.30 Mg C ha^{-1} yr^{-1} by Schuman et al. (2001). When rangeland was grazed rather than excluded from grazing, SOC increased by 0.16 ± 0.12 Mg C ha^{-1} yr^{-1} (Table 6). Establishing grass on previous cropland or reclaimed mineland increased SOC by 0.94 ± 0.86 Mg C ha^{-1} yr^{-1}.

Since SOC could be lost from rangeland at a faster rate than it could be regained, Schuman et al. (2001) suggested that preserving the large reserve of rangeland SOC with good management would be as important for greenhouse gas mitigation as attempting to increase SOC. Follett (2001) estimated that a loss of only 1% of the SOC in the top 10 cm of private grazing lands would be equivalent to the total annual C emission from all cropland in the USA. Given the potential impact of rangelands on C flow, accurate long-term field measurements are urgently needed to quantify the mass, distribution, and dynamics of SOC reserves in rangeland ecosystems.

Soil inorganic C is an important component of many arid and semiarid rangelands soils, but only recently has received attention as a C storage pool (Elbersen et al., 2000). On the shortgrass steppe, grazing management may be affecting SIC more than SOC (Reeder et al., 2004). Total soil C (0–90 cm) was 23.8 Mg ha^{-1} higher with long-term (56 yr) heavy grazing than in an adjacent non-grazed exclosure, with 68% of this difference attributable to SIC and 32% to SOC. Further research is necessary to assess whether the change in SIC in the shortgrass steppe was newly sequestered SIC or a redistribution of SIC within the soil profile as a result of soil water movement with time in response to management and/or precipitation patterns.

3.4. Trace gas flux in rangeland

The amount of information on trace gas flux in rangeland is limited, and often complicated by the high degree of spatial and temporal variability characteristic of native plant communities and their soils. Studies conducted across five rangeland ecosystems in the region have evaluated effects of topography, soil texture, N dynamics, plant species, spatial and temporal variability, burning, and CO_2 enrichment on trace gas flux.

3.4.1. N_2O emission and CH_4 flux

On sand and clay catenas in the Colorado shortgrass steppe, N_2O emission from fine-textured soils at a swale topographic position was greater than emission on coarser-textured soils at a mid-slope position (Mosier et al., 1996a,b) (Table 7). At both topographic positions, N_2O emission increased and CH_4 uptake decreased as soil became wetter. A later study conducted on sites differing in texture but of relatively flat topography showed lower N_2O emission

Table 7
Estimated N_2O emission in rangeland ecosystems

Ecosystem/location	Site characteristics	Experimental features	N_2O emission (g N ha^{-1} d^{-1})	Reference
Shortgrass steppe				
Colorado	Sandy soil	54 month mean flux rate (weekly sampling)	0.30 (mid-slope), 0.49 (swale)	Mosier et al. (1996)
Colorado	Clay soil	54 month mean flux rate (weekly sampling)	0.30 (mid-slope), 0.60 (swale)	Mosier et al. (1996)
Sagebrush steppe				
Wyoming		2 yr (15 sampling dates)	0.88 (*Artemisia tridentat* ssp. *vaseyana*), 0.63 (*A. tridentate*ssp. *wyomingensis*), 0.36 (*Artemisia nova*)	Matson et al. (1991)
Washington		1 yr (monthly sampling)	0.41 (20% was contributed from brief wet periods)	Mummey et al. (1997)
Tallgrass prairie				
Kansas	Variety of slope positions; soils mostly silty clays		18.1 (average flux in wet years) Negligible flux in dry years	Groffman and Turner (1995) and Groffman et al. (1993)
Northern grassland transition zone				
Saskatchewan	Landscape transect; sandy soil	3 yr (monthly or weekly samplings; more often after rainfall events)	0.005 (shoulder; bromegrass dominated), 0.01 (footslope; aspen dominated)	Corre et al. (1996) and Corre et al. (1999)
Saskatchewan	Aspen footslope complex; clay loam soil	1993–1994 (unfertilized), 1994–1995 (fertilized)	0.12, 0.07	Corre et al. (1996) and Corre et al. (1999)
Saskatchewan	Bromegrass shoulder complex; clay loam soil	1993–1994 (unfertilized), 1994–1995 (fertilized)	0.10, 0.13	Corre et al. (1996) and Corre et al. (1999)

but greater CH_4 uptake on the sandier soil, suggesting an important role of texture in controlling N_2O and CH_4 flux (Mosier et al., 1998). Similar results were found in the northern grassland transition zone, where N_2O emission on footslopes was greater than on shoulders (Corre et al., 1995, 1996, 1999). Studies in both ecosystems reflect the importance of topographic position on hydrologic and pedologic processes that ultimately control trace gas flux at the microsite level.

When N was added to the shortgrass steppe, CH_4 uptake was diminished and N_2O emission was enhanced (Mosier et al., 1991). Large N (urea) applications to the shortgrass steppe, either from a single dose (450 kg N ha^{-1}) simulating a cattle urine patch or in smaller doses over a 15-yr period (22 kg N ha^{-1} yr^{-1}), continued to stimulate N_2O emission in both sand and clay loam soils 6–15 yr after application (Mosier et al., 1998). During the first 5 yr after application, N_2O emission remained stable while soil mineral N content declined to 10% of the previous level (Mosier et al., 1996a,b), suggesting that N turnover rather than mineral N pool size affected N_2O emission (Schimel et al., 1989; Davidson and Hackler, 1994).

In the shortgrass steppe, N addition decreased CH_4 uptake in a coarse-textured soil, but was unaffected in a clay loam (Mosier et al., 1996a,b). Valentine et al. (1993) suggested that higher CH_4 flux in mid-slope soils of the catena study cited earlier might be due to lower N turnover and coarser texture. In tallgrass prairie, CH_4 flux was related to soil N status, with CH_4 oxidation inversely proportional to soil mineral N concentration (Chan and Parkin, 2001). Since NH_4 can inhibit CH_4 consumption by methanotrophs and nitrifiers (Bronson and Mosier, 1994), high N turnover rates have been implicated in inhibiting CH_4 oxidation (Mosier et al., 1991; Epstein et al., 1998).

A plant species effect on N_2O emission was observed in a sagebrush steppe ecosystem in Wyoming (Matson et al., 1991), with spring N_2O emission greater under *Artemisia tridentata* ssp. *vaseyana* compared with *A. tridentata* ssp. *wyomingensis* and *Artemisia nova* (Table 7). Plant species also affected trace gas flux in the shortgrass steppe, where Epstein et al. (1998) detected greater CH_4 uptake on sandy clay loam sites dominated by C_4 plants. In the northern grassland transition zone, N_2O emission was greater in an unfertilized aspen complex compared to bromegrass for sandy and clay loam soils (Corre et al., 1995, 1996) (Table 7).

In a shrub-steppe ecosystem in Washington, N_2O emission was greater under *A. tridentata* plants than between plants, expect when dry (Mummey et al., 1997). In their study, spatial variability of N_2O emission ranged from 23 to 130%. However, wetting the soil decreased spatial variability of N_2O emission by releasing and distributing substrates, thereby eliminating hot spots and leading to more uniform soil conditions.

Nitrous oxide and CH_4 emission in rangeland ecosystems is highly variable with time. Twenty percent of the estimated annual N_2O emission occurred within a 48 h period following warm-season precipitation events in a shrub-steppe ecosystem (Mummey et al., 1997). In Saskatchewan, significant in situ emission of CH_4 occurred in spring after snowmelt (80–110 g C ha^{-1} d^{-1}) and in summer (0.7 g C ha^{-1} d^{-1}) at low slope positions following significant rainfall events (Wang and Bettany, 1997). Wintertime flux of N_2O and CH_4 are important in the shortgrass steppe, with 20–40% of annual N_2O emission and 15–30% of annual CH_4 flux occurring in winter (Mosier et al., 1996a,b).

Denitrification was greater in unburned than burned tallgrass prairie (Groffman et al., 1993; Groffman and Turner, 1995), which was attributed to higher soil water content and soil mineral N concentration. Similar year-to-year variation in annual net plant productivity and N_2O emission suggested that both phenomena were controlled by variations in seasonal water and N dynamics. A strong relationship with normalized difference vegetation index (NDVI) allowed a landscape-level estimation of N_2O emission over the Konza Prairie site of 18.1 g N ha^{-1} d^{-1} in a wet year (Groffman and Turner, 1995), and negligible emission in dry years (Table 7).

Carbon dioxide enrichment of the atmosphere over shortgrass steppe vegetation had no significant effect on N_2O and CH_4 flux (Mosier et al., 2002), even though one of the main effects of elevated CO_2 was increased plant water potential and soil water content (Morgan et al., 2001, 2004; Nelson et al., 2004).

3.4.2. Soil CO_2 emission

Unlike N_2O and CH_4, the land–atmosphere exchange of CO_2 has been evaluated at two important levels, the soil, and the stand or ecosystem level. The ecosystem level integrates both soil and plant emission and assimilation into plant biomass during the day, whereas soil measurements consider microbial and plant root emission only. Thus, the two levels of CO_2 flux are considered separately since the underlying processes, rates, and direction of flux differ.

Exposure of shortgrass steppe to elevated CO_2 (720 μmol mol^{-1}) in open top chambers (Morgan et al., 2001, 2004) did not affect soil CO_2 emission (Mosier et al., 2002), which averaged 3.9 Mg C ha^{-1} yr^{-1} (Table 8). However, Pendall et al. (2004) found that soil CO_2 emission in elevated CO_2 chambers was 25% greater than in ambient CO_2 chambers in a moist growing season, and 85% greater in a dry growing season. Annual soil CO_2 emission was 3.5 and 4.6 Mg C ha^{-1} yr^{-1} under ambient and elevated CO_2 conditions, respectively (Table 8). Stable C isotope partitioning between decomposition and rhizosphere respiration indicated that soil CO_2 enhancement with elevated CO_2 was due to higher decomposition rate from increased soil water content and increased substrate availability (Pendall et al., 2004).

Soil CO_2 emission in a mixed-grass prairie was low in spring and autumn, intermediate in late spring, and highest in early June at the time of maximal aboveground biomass production (Frank et al., 2002). Growing-season soil CO_2 emission averaged 3.5 g C m^{-2} d^{-1} when ungrazed and 4.3 g C m^{-2} d^{-1} when grazed. Over the 3-yr study, growing-season soil CO_2 emission averaged 7.3 Mg C ha^{-1} and dormant-season emission was 0.9 Mg C ha^{-1} (Table 8).

Soil CO_2 emission in annually burned tallgrass prairie exceeded that in unburned prairie by 20–55%, with the largest difference occurring in April (Table 8). Soil CO_2 emission ranged from 13 Mg C ha^{-1} yr^{-1} in unburned sites to 21 Mg C ha^{-1} yr^{-1} in annually burned and irrigated tallgrass prairie sites (Knapp

Table 8
Estimated annual soil CO_2 emission in rangeland ecosystems

Ecosystem/location	Soil classification	Textural class	Scale	Experimental features	Annual CO_2 emission (Mg C ha^{-1} yr^{-1})	Reference
Shortgrass steppe						
Colorado	Ustic Haplocambid	FSL	Soil	54 month mean flux rate (weekly sampling) from chamber method	3.8 (ambient CO_2), 3.9 (elevated CO_2)	Mosier et al. (2002)
Colorado	Ustic Haplocambid	FSL	Soil	Fluxes measured periodically over 2 yr with soil gas sampling tubes	3.5 (ambient CO_2), 4.6 (elevated CO_2)	Pendall et al. (2004)
Mixed-grass prairie						
North Dakota	Entic Haplustoll, Typic Haplustoll, Typic Calciustoll	L, SiL, SiCL	Soil	Flux measured at 21-d intervals during growing season, and periodically during dormant season for 4 yr with small chambers	8.2	Frank et al. (2002)
Tallgrass prairie						
Kansas	Udic Argiustoll	SiCL	Soil	Fluxes measured 7–14 d intervals during growing season, and 15–30 d intervals during dormant season for 2 yr with dynamic-chamber method	13 (unburned), 16 (infrequent fire), 19 (annual fire), 21 (annual fire and irrigation)	Knapp et al. (1998)
Kansas	Udic Argiustoll	SiCL	Soil	Fluxes measured weekly to monthly over 1 yr with dynamic-chamber method	13 (non-defoliated), 11 (early-season defoliated), 11 (full-season defoliated)	Bremer et al. (1998)

Table 9
Estimated annual whole system CO_2 flux in rangeland ecosystems

Ecosystem/location	Soil classification	Textural class	Scale	Experimental features	Estimated annual CO_2 flux[a] (Mg C ha^{-1} yr^{-1})	Reference
Sagebrush steppe						
Idaho	Typic Calcixeroll, Pachic Haploxeroll, Pachic Argixeroll	SCL, L	Whole system	Bowen ratio energy balance measurements of CO_2 fluxes conducted across 4 yr; modelling for gap-filling and dormant season fluxes	1.0	Gilmanov et al. (2003)
Mixed-grass prairie						
North Dakota	Entic Haplustoll, Typic Haplustoll, Typic Calciustoll	L, SiL, SiCL	Whole system	Bowen ratio energy balance measurements of system CO_2 flux conducted for three growing seasons, plus chamber measurements of dormant season respiration	−0.2 (loss) to 0.5 (uptake) for ungrazed prairie	Frank (2002)
Tallgrass prairie						
Oklahoma	Pachic Argiustoll	SiCL	Whole system	Eddy covariance fluxes measured continuously through 3 yr on annually grazed pastures; interpolation and modelling for gap filling	0.5–2.7 (C lost in burning unaccounted for)	Suyker and Verma (2001)

[a] +, assimilation; −, emission.

et al., 1998). Soil CO_2 emission was 13, 11, and 11 Mg C ha^{-1} yr^{-1} for non-defoliated, early-season defoliated, and full-season defoliated tallgrass prairie (Bremer et al., 1998). Grazing of tallgrass prairie by bison reduced soil CO_2 emission by as much as 30% compared with non-grazed pastures (Knapp et al., 1998). A large portion of the variability in soil CO_2 emission was accounted for by variations in temperature, water-filled pore space, and plant growth (Knapp et al., 1998; Franzluebbers et al., 2002).

3.4.3. Whole-ecosystem CO_2 flux

LeCain et al. (2002) found little effect of grazing intensity on seasonal mean mid-day whole-ecosystem CO_2 flux, with rates ranging from 2.1 to 3.5 kg C ha^{-1} h^{-1} and maximal values of 8.4 kg C ha^{-1} h^{-1} at peak plant biomass accumulation and high soil water content. Similar results were obtained under both shortgrass and mixed-grass prairies, suggesting that livestock grazing at recommended stocking rates had minimal impact on annual C assimilation potential (LeCain et al., 2000, 2002).

Growing season whole-ecosystem CO_2 flux in a sagebrush steppe varied from 0.8 to 3.0 Mg C ha^{-1} yr^{-1} due to variations in precipitation amount and distribution (Gilmanov et al., 2003). A mean value of 1.0 Mg C ha^{-1} yr^{-1} was estimated for the 4-yr period (Table 9). The CO_2 flux data were highly related to time-integrated Advanced Very High Resolution Radiometer NDVI (Wylie et al., 2003). Using eddy covariance from an aircraft 60–90 m above ground level to measure CO_2 flux over a sagebrush steppe, there was a positive linear relationship with absorbed photosynthetically active radiation (Kelly et al., 2002). These studies indicate considerable promise for using remote sensing to scale CO_2 fluxes to landscape and regional levels.

Whole-ecosystem CO_2 flux in a mixed-grass prairie was near zero or slightly negative in early spring, increased as leaf area index increased, declined in late summer as plants senesced, and reached zero or became negative in autumn (Frank and Dugas, 2001; Frank, 2002). Growing season whole-ecosystem CO_2 flux ranged from 0.7 to 0.9 Mg C ha^{-1}, but due to dormant-season emission annual CO_2 flux ranged from −0.2 to 0.5 Mg C ha^{-1} yr^{-1} (Table 9). Frank (2002) concluded that more research was needed to understand dormant-season CO_2 flux, since yearly variation was high and often determined whether grasslands were sources or sinks for C.

Whole-ecosystem CO_2 flux in a tallgrass prairie was greater under elevated CO_2 (31 kg C ha^{-1} d^{-1}) than under ambient CO_2 (25 kg C ha^{-1} d^{-1}) (Ham et al., 1995). This difference was attributed to improved plant water relations and a longer growing season under elevated CO_2.

In an annually burned but non-grazed tallgrass praire, annual whole-ecosystem CO_2 flux ranged from 0.5 to 2.7 Mg C ha^{-1} yr^{-1} (Table 9) (Suyker and Verma, 2001; Suyker et al., 2003). Seasonal rates in whole-ecosystem CO_2 flux increased with leaf area index with a maximum of 84 kg C ha^{-1} d^{-1} in mid-July (Suyker and Verma, 2001). Considering the C lost from burning, this tallgrass prairie was at equilibrium with respect to annual C exchange.

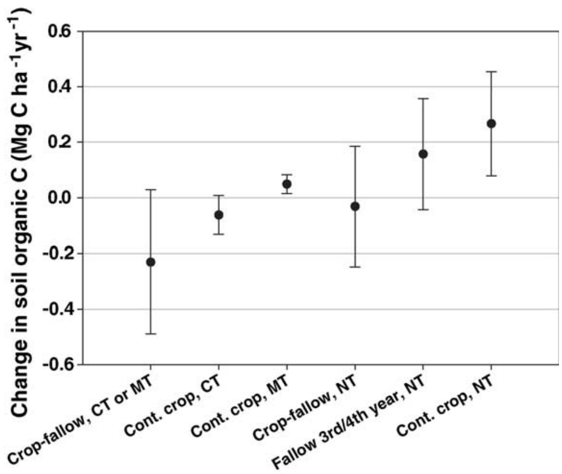

Fig. 4. Accrual/loss rates of soil organic carbon (SOC) for six dryland cropping systems predominant within study region, ±one standard error.

4. Synthesis

The region reviewed in this manuscript includes agricultural land recognized throughout North America for its inherently fertile soils, which contribute to the production of large quantities of grain and forage in spite of highly variable climatic conditions. Economically and environmentally, the importance of the region cannot be overstated. Gross farm receipts in the region exceeded $65 billion in 2000, representing approximately 9% of the Gross Domestic Product for the states and provinces included in the region (Statistics Canada, 2003; Bureau of Economic Analysis, 2004). Environmentally, the region is important by virtue of its size and prevalence of cropland and rangeland, the management of which affects environmental quality at local, regional, and global spatial scales.

The purpose of this review was to provide an accounting of past research concerning management effects on C sequestration and N_2O and CH_4 flux in cropland and rangeland in the northwestern USA and western Canada. Given the multifaceted topic and the expansive region, no assurances can be made that all applicable literature was included in the review. Furthermore, the broad topic did not lend itself well to a few comprehensive conclusions. As a substitute, a synthesis of each section is provided below, with reference to IPCC coefficients and suggestions for future research.

Crop management effects on SOC were found to be highly variable within the delineated region, ranging from −0.53 to 1.68 Mg C ha^{-1} yr^{-1}. Among dryland cropping systems, continuous cropping under NT resulted in consistent increases in SOC, averaging 0.27 ± 0.19 Mg C ha^{-1} yr^{-1} (Fig. 4), a value similar to the estimated net annual change in C stocks from improved cropland management in Annex I countries (0.30 Mg C ha^{-1} yr^{-1}) (IPCC, 2000). Minimum tillage with continuous cropping also resulted in consistent increases in SOC, but at a much lower rate (0.05 ± 0.03 Mg C ha^{-1} yr^{-1}) (Fig. 4). Inclusion of fallow in rotation with NT resulted in highly variable C sequestration rates, indicating the need for further research to explore mechanisms underlying C accretion (or depletion) for this management system. Inclusion of high residue-producing crops or legume green manures in rotation increased SOC by 0.10–0.22 Mg C ha^{-1} yr^{-1}. Low residue-producing crops (e.g., flax) in rotation, however, decreased SOC. Greater adoption of alternative crops by producers in the region (Tanaka et al., 2002; Padbury, 2003) warrants the need for further research on their long-term effects on SOC. Nitrogen fertilization increased SOC throughout the region in continuously cropped management systems. Irrigated, continuous crop, NT or MT management systems were the most effective means to increase SOC on cropland in the region

(1.56 ± 0.17 Mg C ha^{-1} yr^{-1}; $n = 2$), although limited research has been conducted in these systems.

Estimates of C sequestration potential for rangelands were more difficult to attain than for cropland, because rangelands occupy a greater diversity of plant communities, soils, and landscapes. This diversity increased spatial and temporal variability of SOC. Despite high variability, grazing was estimated to increase SOC storage by 0.16 ± 0.12 Mg C ha^{-1} yr^{-1}. The rate of SOC storage following establishment of grass on previous cropland or reclaimed mineland was 0.94 ± 0.86 Mg C ha^{-1} yr^{-1}. A rate of 0.50 Mg C ha^{-1} yr^{-1} has been estimated by the IPCC for improved grazing land management in Annex I countries (IPCC, 2000). Increases in SOC stocks in rangelands appeared to be most closely associated with shifts in plant communities resulting in either greater plant biomass production, increased root biomass, or introduction of legumes into the system.

Assessments of trace gas flux from cropland and rangeland were lacking for much of the region, with most research concentrated in northeastern Colorado. The few evaluations conducted indicate emission of N_2O was lowest from rangeland (≤ 1.0 g N ha^{-1} d^{-1}) and greatest from irrigated cropland (5.1–56.1 g N ha^{-1} d^{-1}), but highly variable in all agroecosystems (%CV = 42–105%) (Table 10). Differences in N_2O emission among rangeland ecosystems appeared to be driven by water, N, and plant community dynamics. Water and N were also important drivers of N_2O emission in cropping systems, and when combined, resulted in the highest N_2O emission among all agroecosystems evaluated in the region.

Typically an IPCC 'emission factor' of 1.25% would be applied to annual soil N inputs to estimate direct N_2O emission from soils (i.e., 1.25 kg N_2O-N emitted per 100 kg N input) (Bouwman, 1994). The N inputs may include inorganic fertilizers, manure and other organic amendments, crop residue N, as well as biologically fixed N. For studies reported in this review, the emission factor for N_2O was $1.3 \pm 0.8\%$ for all cropland treatments, and $1.1 \pm 0.5\%$ for cropland without application of cattle manure or sewage sludge.

Rangeland and non-irrigated cropland both appeared to act as a sink for atmospheric CH_4, but conclusions regarding agroecosystem effects on the flux of this important greenhouse gas are at best tenuous. Additional research is needed to quantify both N_2O and CH_4 flux from the broad portfolio of rangeland ecosystems and cropping systems present throughout the region.

Table 10
Mean N_2O emission for rangeland and cropland within study region

Agroecosystem	N_2O emission (g N ha^{-1} d^{-1})	CV (%)[a]	n[b]
Rangeland			
Northern grassland transition zone	0.07	71	6
Sagebrush steppe	0.57	42	4
Shortgrass steppe	1.00	105	8
Cropland			
Non-irrigated, 0–50 kg N ha^{-1}	2.6	85	8
Non-irrigated, >50 kg N ha^{-1}	3.7	51	10
Irrigated, 0–50 kg N ha^{-1}	5.1	104	9
Irrigated, >50 kg N ha^{-1}	11.1	69	13
Irrigated with manure/sludge	56.1	103	5

[a] Coefficient of variation.
[b] Number of values used to calculate mean.

Rates of soil and whole-ecosystem CO_2 flux from cropland indicated short-term C loss/gain from specific management practices mirrored long-term changes in SOC. Among rangeland ecosystems, soil CO_2 emission appeared related to productivity, with emission of approximately 3.5–4.5 Mg C ha^{-1} yr^{-1} in the shortgrass steppe, approximately 8.2 Mg C ha^{-1} yr^{-1} in the mixed-grass prairie, and 10–20 Mg C ha^{-1} yr^{-1} in the tallgrass prairie. Assessment of whole-ecosystem CO_2 flux indicated rangelands were generally a net sink for C, ranging from -0.2 to 2.7 Mg C ha^{-1} yr^{-1}.

Acknowledgments

We thank Al Frank, Rebecca Phillips, and four anonymous reviewers whose comments improved earlier drafts of the manuscript.

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